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Transport Non-exhaust PM-emissions

An overview of emission estimates, relevance, trends and policies

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Summary

Exhaust PM emissions have been decreasing over the last 20-30 years, under impulse of targeted policy interventions. Meanwhile, non-exhaust PM emissions from transport are steadily increasing on par with increasing transport demand. In particular sources of non-exhaust PM emissions from road transport from brake, tyre or road wear have all increased, and non-exhaust PM emission have overtaken exhaust emissions as the dominant emission source in transport as of 2012 for PM_{10} and 2018 for $PM_{2.5}$. Also for rail and aviation, there is emerging evidence on the importance of non-exhaust emissions.

Rail, aviation and in particular road transport are a source of non-exhaust PM emission, associated with tyre, brake and road surface wear and tear. While exhaust emissions from transport are under control and steadily declining as a results of ever stringent fuel standards and emission standards, non-exhaust PM-emission are increasing as transport demand increases, without any policies in place to specifically reduce non-exhaust PM-emissions.

At the policy level, historically the focus has almost exclusively been on reducing exhaust-emissions. Few policy interventions currently exist that affect non-exhaust emission directly. Technical solutions are available to substantially reduce non-exhaust PM, in particular for road brake and tyre wear. A variety of regulatory options are available such as emission limits, tyre and brake standards.

Estimating non-exhaust PM emission is subject to high uncertainty, due to the lack of up-to-date emission inventory guidelines and lack of research. By all estimates we found, non-exhaust PM-emission from transport have already overtaken exhaust PM-emissions in importance, first for PM10 and more recently for PM2.5 as well. Non-exhaust PM-emission thus require a stronger policy focus, especially as there is emerging evidence wear and tear is an important source for microplastics in the environment.

Technological options are available to reduce the impact and policy makers have a variety of regulatory options available to tackle the issue of non-exhaust PM, in the same way as exhaust emissions from transport has been addressed in the past decades.

The report gives a comprehensive literature review on the non-exhaust PM emission from transport. All types of wear particles are considered (brake, tyre, road surface) and all modes (road, rail, aviation), with strong emphasis on road. The report serves as an input to review current emission inventories, summarizing the current emission estimates, the estimation methodologies, uncertainties and future trends, briefly zooming in on the relevance of electric vehicles. The report considers both air quality as well as the relevance of non-exhaust emission as a source of microplastics. To conclude, the report includes a brief overview of technological and policy options to reduce the environmental impact.

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1 Context

Human exposure to particulate matter (PM) concentrations in ambient air is a well-known health concern. Particulate matter directly emitted to the air (primary PM emissions) are typically categorized as exhaust and non-exhaust emission. Exhaust PM emissions are the product of combustion processes, released as a result of incomplete combustion of fossil fuels. Non-exhaust emissions however are the result of friction and are linked to tyre, brake and road wear.

EU air pollution legislation has targeted exhaust emissions in the transport sector primarily, imposing fuel quality standards and vehicle emission standards (Euro standards) to reduce exhaust emissions. As a result, exhaust primary PM emissions have been decreasing over the last decades, while unregulated non-exhaust PM emissions have been increasing with increasing road transport activities. The growing relative importance of non-exhaust PM emissions implies more focus on mitigating such emissions.

Primary PM emissions are currently reported under the Convention on Long-range Transboundary Air Pollution (CLRTAP), and the EU's National Emission Reduction Commitment Directive (NEC Directive). The NEC Directive transposes the reduction commitments for main air pollutants for 2020 agreed by the EU and its Member States under the 2012 revised Gothenburg Protocol (CLRTAP) and sets commitments for 2030, too.

Under the NEC Directive, EU Member States shall prepare national emission inventories, including primary PM emissions, using the methodologies adopted by Parties to the CLRTAP (EMEP Reporting Guidelines). Further, the Member States are requested to use the EMEP/EEA air pollutant emission inventory Guidebook (EMEP/EEA Guidebook) referred to in the Guidelines (Table 1).

In 2021, EEA plans to update the sections in the EMEP/EEA Guidebook addressing non-exhaust emissions from transport. One aim of this report is also – as far as feasible - to contribute to an update of the Guidebook, respectively.

Table 1: Overview of current air emission reporting obligations in the EU

Legal obligation	Emissions to report	Annual reporting deadline for EU Member States	Annual reporting deadline for the EU (ª)
LRTAP Convention (ᵇ)	NO_x (as nitrogen dioxide — NO_2), NMVOCs, SO_x (as SO_2), NH_3, CO, HMs, POPs and PM	15 February 2020	30 April 2020
NEC Directive	NO_x (as NO_2), NMVOCs, SO_x (as SO_2), NH_3, CO, HMs, POPs and PM	15 February 2020	Not applicable
EU MMR/United Nations Framework Convention on Climate Change (UNFCCC)	Carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulphur hexafluoride, NO_x, CO, NMVOCs and SO_2	15 January 2020 to the European Commission and 15 April 2020 to the UNFCCC	15 April 2020

Notes: (ª) Over the years, the European Community and the EU have signed a number of protocols. The commitments include varying numbers of EU Member States. Therefore, emissions must be reported separately for the EU-9, EU-12, EU-15, EU-27_2007 and EU-28 (see Table A2.2 in Appendix 2 for more information on EU country groupings).

(ᵇ) Parties are formally required to report only on the substances and for the years set forth in protocols that they have ratified and that have entered into force.

Source: EEA, 2020c (LRTAP report).

The emergence of electric vehicles increases the relevance of non-exhaust emissions even further. While these vehicles reduce the exhaust emissions to zero([1]) and thus promise to reduce total emissions in the transport sector even further, electric vehicles do not eliminate non-exhaust emissions.

There is also an increasing concern that non-exhaust PM, in particular released through tyre wear, is potentially an important source of heavy metals and microplastics to the environment.

This paper reviews the current knowledge on sources of non-exhaust particulate emissions from transport, on emission trends and on the relevance of non-exhaust PM emissions containing microplastics. Finally, it looks into existing and emerging technical and regulatory options to reduce non-exhaust PM emissions.

Box 1: Terminology regarding PM wear and microplastic

The terminology and abbreviations used in this paper: * **WP**: Wear particles. * **MP**: Microplastic Particles. * **TP**: Tread Particles. Tread rubber particles that result from the wearing off process and use of tyres. * **TWP**: Tyre Wear Particles. Particles produced by wearing off tyres by the shear forces between the tyre and the surface. TWP contains mineral incrustation of the pavement into the TP. * **RWP**: Road Wear Particles. Particles that originate at the road surface as a result of wearing off processes. * **TRWP**: Tyre and Road Wear Particle, when it contains the tread rubber and embedded road material. * **BWP**: Brake Wear Particles. Particles that result from the wearing off process of vehicle brakes during braking processes. * **NEE**: Non Exhaust Emissions. A combination of all particles generated, excluding exhaust emissions.

[1] "The evidence base on electric vehicle life cycle analysis (LCA) impacts needs to continue to be updated and eveloped. It should reflect the different electric vehicle makes and models increasingly available, emerging data on real-world use and how batteries are treated at the end-of-life stage. Furthermore, there is a need to ensure that the studies continue to account for full LCA impacts rather than the historical focus on GHG emissions." (EEA, 2018, TERM report : https://op.europa.eu/en/publication-detail/-/publication/c2046319-0731-11e9-81b4-01aa75ed71a1/language-en.

2 Sources and estimation efforts

Non-exhaust PM emissions are the result of friction, releasing material of the friction surfaces. In road transport, the wearing processes associated with traffic predominantly affects the road surface and the tyres and brakes of the vehicles. Emissions are estimated by capturing generated particles either in real world environments or in a simulated environment in a laboratory, such as pin-on-disc studies for brake wear (Kukutschová and Filip 2018) and road simulators for tyre wear (Gustafsson 2017). Another approach is to estimate weight loss of brakes and tyres as an estimate for volumes released to the environment. However, only a fraction of the abrasion material is released as PM (Baensch-Baltruschat 2020). In this paper, we distinguish between wear particles and microplastic particles.

The **wear particle** (WP; see Box 1) size ranges from a few nm to several 100's of µm. For Air quality, the primary focus is on particles with a size below 10 µm (PM_{10}) and 2.5 µm ($PM_{2.5}$). Only PM with an aerodynamic diameter smaller than about 10-20 µm will have terminal velocity slow enough to be airborne for a significant amount of time and thus have the potential to be dispersed by air. Suspension of PM can be wind-driven, but the more important process is turbulence around the driving vehicles. On the road, the lifting forces that are strong enough to do the initial lifting of PM, are centred around the tyres. Suspension and resuspension are thus important factors in the relevance of WP for ambient air quality, but not for formation of non-exhaust PM itself (see details in Box 2).

Once **microplastic particles** (MP; see Box 1) are in the atmosphere, they are subject to the same forces as all suspended particles. Particles move with the air flows until they are removed. MP can be removed again from the atmosphere by gravitational settling, dry deposition or by precipitation. Most road wear particles with MP will have hydrophobic properties, thus their removal will be relatively less efficient by precipitation, which is the most efficient removal mechanism for most PM (Grythe et al., 2016).

Box 2: Relevance of resuspension

Resuspension is not a source of non-exhaust emissions. However, to determine air quality levels, not only the newly generated particles (i.e. emissions) contribute but also the resuspended particles that settled, yet become airborne again due to wind and passing vehicles. Resuspension is contributing to poor air quality but is not a primary source of emissions.

Resuspended particles cannot per se be attributed to BWP or TRWP exclusively; it is a mix of resuspended PM from road transport sources (exhaust and non-exhaust) as well as any other dry deposition from other sources such as construction sites. (Air quality expert group 2019, Denby 2018). This poses a risk of double counting in an emission inventory.

Resuspension is an important source of poor air quality. Resuspension is a high contributor to total PM-contribution from road transport, 50-75% according to Timmers and Achten (2016) using emission factors from Amato (2012) who report a wide bracket of emission factors associated with resuspension, namely 9.4-36.9 mg/km PM_{10}. A review by Denby (2018) shows only 5-40% of these particles to be in the $PM_{2.5}$ fraction. The importance of resuspension is confirmed by Van Der Gon (2018), concluding that 'resuspension of road dust may well be the dominant source of road transport PM_{10} in many cities', and recommending to include resuspended PM sources as a separate Nomenclature For Reporting (NFR) category for emission inventory compilation under the Gothenburg Protocol (LRTAP Convention) and NEC Directive.

Currently, The EEA/EMEP Guidebook does not include resuspension as a primary emission source.

The next sections look into the sources of non-exhaust PM emissions in further detail, focussing primarily on road transport, but also briefly discuss sources from rail and aviation.

2.1 Road transport

There are three main types of non-exhaust PM emissions from road transport: brake-wear particles, tyre-wear particles and road-wear particles. These have different formation pathways and have different characteristics.

2.1.1 Brake-wear particles (BWP)

Brakes use friction to generate braking power. By forcing a brake pad and a rotating disc or drum together, the generated friction causes the vehicle to reduce speed and unavoidably causes abrasion both of the brake pad and of the surface of the disc or drum. The abrasion leads to PM emitted directly into the air.

When braking, the strongest frictional forces are on the brake pads. Modern vehicles are usually equipped with non-asbestos organic (NAO), semi-metallic or low metallic brake pads, which differ in performance and durability (see Kukutschová and Filip, 2018). For a detailed description of the brake components and composition see Grigoratos and Martini (2014). Brake linings generally consist of binders, fibres, fillers, frictional additives or lubricants, and abrasives at proportions that depend on the type of lining and the manufacturer (see Grigoratos and Martini, 2014 and references therein for details). Binders, used to hold the components of the brake pad together, consist of a variety of modified phenol-formaldehyde resins. The fibres, which constitute around 6-35% of the mass of the lining, are used as reinforcement and they can be metallic, mineral, ceramic or organic, and mainly consist of copper, steel, brass, potassium titanate, glass, organic material (aramid) and Kevlar®[2]. The fillers are used to improve the brake pad properties and tend to be barium and antimony sulphate, magnesium oxides and chromium oxides, silicates, ground slag, stone, and metal powders. The additives or lubricants, which make around 5-29% of the brake lining mass, can be inorganic, metallic or organic. Common materials are graphite, ground rubber, metallic particles, black carbon, cashew nut dust, and antimony trisulphide. Abrasives are around 10% of the lining mass, and aluminium oxide, iron oxides, quartz and zircon are typically used.

We find a wide range of emission factor values for BWP. The variation is attributed to different brake material being used, the testing methodology and test cycles. Under the Worldwide Harmonised Light Vehicles Test Procedure (WLTP) in a wind tunnel, Athanasios (2019) finds an emission factor of 10 mg/km PM_{10}([3]) (5 mg/km per brake), 30% of which is $PM_{2.5}$ ([4]). On-road vehicle measurements conclude a range of 2.8-4.1 mg/km (1.4-2.1 mg/km per brake) for PM_{10} under the Los Angeles City Traffic (LACT) driving cycle (Ferdinand 2019). These recent findings are in line with older ones (Hulskotte 2014) and the existing EMEP/EEA Guidebook ranging 4.4-10 mg/km for PM_{10} and 1.7-3.9 mg/km for $PM_{2.5}$ (Ntziachristos and Boulter 2019).

The material at the basis of the BWP emission generation is 'a multicomponent composite, typically formulated of more than 10 constituents bound in a polymer matrix' (Massimo and Annunziata 2018). These materials include organic compounds from fibres and metal oxides from the abrasive material in the linings but also metal oxides from the brake disc. The chemical composition of BWP has been addressed in different studies and summarised in literature reviews (Grigoratos and Martini, 2014;

[2] Kevlar® lightweight synthetic fibre of exceptionally high strength and heat resistance.

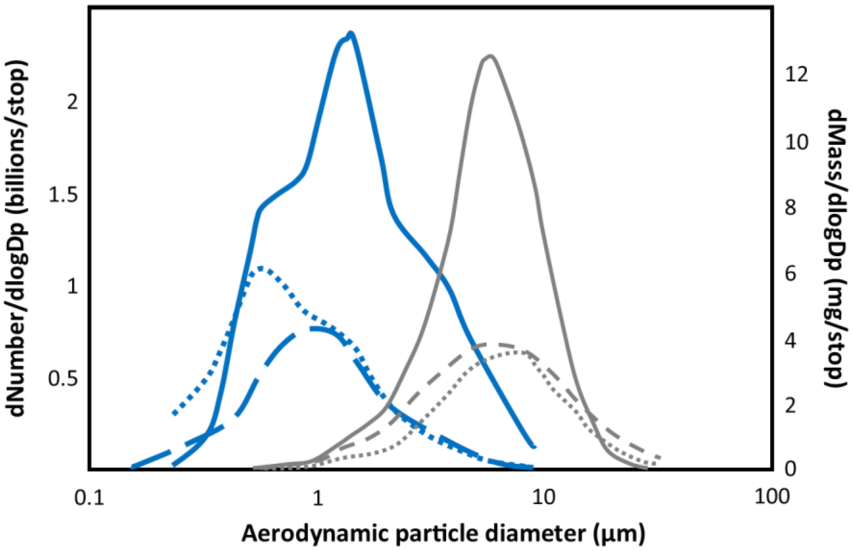
[3] PM_{10} = fraction of particles with an aerodynamic diameter smaller than 10 µm.

[4] $PM_{2.5}$ = fraction of particles with an aerodynamic diameter smaller than 2.5 µm.

Kukutschová and Filip 2018; Hulskotte 2014). Most of the studies report iron (Fe), copper (Cu), zinc (Zn), zirconium (Zr), tin (Sn) and antimony (Sb) as abundant heavy metals, and thus have been commonly used as tracers for BWP (Pant and Harrison, 2013 and references therein). Carbonaceous species have also been identified as constituent of BWP in addition to the metals, such as black carbon and graphitic material (Kukutschová et al. 2011) and organic, elemental or total carbon (Gasser et al. 2009; Garg et al., 2000). Moreover, organic compounds (e.g., polyalkylene glycol ethers, n-alkanoic acids, PAH-polycyclic aromatic hydrocarbons) have been identified, however, little information is available (Grigoratos and Martini, 2014).

In a meta-analysis Grigoratos and Martini (2015) find agreement on the size distribution of BWP, unimodal distributed with most particles about 2-6 µm in size, and they estimated that about 50% of all BWP generated will become airborne. The generation of BWP tends to increase as the temperature of the brake increases, associated with stronger braking action which is more common in urban driving conditions (Athanasios 2019). Figure 1 illustrates the results published by Sander et al. (2003) on the mass and number size distribution of BWP for the three different types of brake pads, i.e., low-metallic, semi-metallic and NAO where NAO is the most important type as a source of MP.

Figure 1: Sketch showing the number (blue) and mass (grey) distribution of BWP from low-metallic (line), semi-metallic (dotted line) and NAO (dashed line) brake pad materials



Source: modified after Sanders et al. (2003).

2.1.2 Tyre-wear particles (TWP)

The tyre surface is steadily abraded by contact with the road surface, generating particles of mixed size. While PM of large size will typically remain on the road surface, PM of smaller size range can become airborne contributing to PM in the atmosphere.

The structure and composition of tyres has changed over time. Initially tyres were made of natural rubber, whereas nowadays they are a mixture of natural and synthetic rubbers (petroleum polymers) with other additives. An overview by Kole et al. (2017) shows that a variety of different compounds are added to improve the properties of the rubber tyre. Hereby, sulphur (1-4%) is added to vulcanise the rubber and obtain a highly elastic material, zinc oxide (1%) as a catalyst and black carbon (22–-40 %) as a filler and to make the tyre UV-resistant. Over time, these additives have also been modified, e.g., carbon has been partially replaced by silica, reducing the road resistance. Furthermore, oils are added to make the tyre more flexible and to improve performance.

TWPs are for the most part not pure but contain mineral incrustation of pavement into the thread particles (TP; see Box 1). The chemical composition of TWP and TP have been compared and evaluated in previous studies (Kreider et al., 2010; Panko et al., 2018). The main chemical difference between TWPs and TP is that TWP contain encrustations of mineral, producing and enrichment in crustal elements (e.g., Aluminium (Al), Magnesium (Mg), Fe, Potassium (K)) and depleting key elements of the tyres (i.e., polymers, black carbon, Zn, Polycyclic Aromatic Hydrocarbon (PAH)). In real world conditions, tyre wear particles are always a mix of material from tyres and the road surface (Kreider 2010). In this sense, (Baensch-Baltruschat 2020) proposes the term Tyre and Road Wear Particles (TRWP; see Box 1) as a more suitable definition and categorization.

TRWP consist mainly of rubber hydrocarbons, carbon black[5] and silicon (Si) from the tyre tread. This PM includes traces of compounds of vulcanization both organic and inorganic in nature (Grigoratos and Martini 2015). Only a limited fraction of TRWP will become airborne and contribute to PM air pollution. The airborne fraction is considered not to exceed a maximum of 10 % of the total tyre wear material (Grigoratos and Martini 2015).

The chemical composition of "pure" TWP, including both crustal material and TP, and inconsistencies in studies regarding their mass distribution, make the identification of microplastics in airborne PM challenging. Existing standards define common methodologies to identify tyre wear particles and which markers can be used for the identification. These standards, such as the ISO Technical Specification (TS-20593), establish principles for air sample collection and quantification of generated polymers. In addition, they establish chemical markers (butadiene, isoprene and styrene) for identification of tyre wear particles.

Baensch-Baltruschat (2020) provides an extensive overview of the emission factor estimation, making a clear distinction between [1] the total material released into the environment, including the coarse fraction that is emitted to soil and water and [2] the airborne fraction, both quantified in mg/km. Values for the airborne fraction range between 2 and 8 mg/km. The current EMEP/EEA Guidebook distinguishes between tyre wear and road surface wear. The TIER 2 emission factors for total suspended matter (TSP) are 6.7-16.2 mg/km for the former, of which 60% is PM_{10} and 15 mg/km for the latter, of which 50 % is PM_{10} (Ntziachristos and Boulter 2019).

Kreider et al. (2010) reported less than 1% by volume of the particles were less than 10 µm. However, in an unpublished research by Palko, 60% of TWP (by mass) were present in the $PM_{2.5\text{-}10}$ fraction. In a literature review, Grigoratos and Martini (2014) list a number of studies placing the mass fraction within the PM_{10} range to span an order of magnitude (0.84-8.5%). The review further shows that studies find both unimodal and bimodal mass distributions, where the presence of a second finer mode is the main difference between studies. The wide range of share of PM_{10} to total TWP can in part be explained by the presence of such a mode. The difference between unimodal and bi-modal TWP particle distribution has been attributed to differences in the tyre type (e.g., studded and non-studded), type of pavement, driving conditions or vehicle load (Kupiainen et al., 2005; Gustafsson et al., 2008) However, according to the review by Grigoratos and Martini (2014) the discrepancies may well be attributed to the different analytical techniques employed in the studies, which make the comparison between different studies difficult. Based on the available studies, it is possible to conclude

[5] Carbon black is distinguished by a high EC content and well-controlled properties whereas Black carbon (BC) particles are characterized by their heterogenous properties (Long 2013).

that there are large uncertainties regarding the size of TWP and whether they are small enough to be suspended.

2.1.3 Road-wear particles (RWP)

The road surface is a combination of asphalt, a mixture of aggregates and bitumen. Bitumen is a petroleum product typically enhanced by polymers (Porto et al. 2018), whereas aggregates are typically crushed rocks or gravel, and thus of crustal composition. In addition, many road surfaces have road paint, which is predominantly a thermoplastic paint, that is also exposed to wear.

As indicated in the previous section, RWP and TWP combine to form TRWP in real world environments (Baensch-Baltruschat 2020). The term RWP refers to particles originating from the road surface. These are particles of fragmented road pavement, formed as a consequence of the interaction with the vehicle tyres. The asphalt and road paint are the main surface of RWP.

The generation of particles is linked to the road type (Penkala 2018). Clearly, unpaved roads generate more particles than paved roads, but also the composition of paved roads is likely to influence the generation of particles. Penkala (2018) proposes the following typology: susceptible surfaces (like gravel), semi-rigid (asphalt surface with a concrete base) and rigid (cement concrete surface).

RWP can also include resuspension of wind blow particles originating elsewhere, in addition to ground sanding and salt particles used on the road surface (Denby et al., 2013).

2.1.4 Road transport emission estimation efforts

EU Member States currently have a reporting obligation through the EU's NEC Directive. The European Monitoring and Evaluation Programme (EMEP) has developed guidelines to produce national emission inventories. Non-exhaust emissions from road transport are included as two separate NFR categories:

- 1.A.3.b.vi Road transport: Automobile tyre and brake wear
- 1.A.3.b.vii Road transport: Automobile road abrasion.

Although the emission inventory methodology was briefly reviewed in 2019 (Ntziachristos and Boulter 2019), it has not changed substantially over the past 10 to 15 year. Comparing the chapter's version of 2009, 2013, 2016 and 2019, the methodology has not changed and is based on old references, 90% of which are dated before 2005. The only significant change was implemented in 2016 when the emission factors were updated after the publication of Grigoratos and Martini (2015). The fact that the EMEP/EEA Guidebook combines tyre and brake wear and considers road abrasion as a separate category is in conflict with recent finding considering tyre wear and road abrasion as a single species in real world conditions. (Baensch-Baltruschat 2020).

The EMEP/EEA Guidebook proposes a 2 TIER approach to estimate non-exhaust PM. The TIER 1 approach proposes fixed emission factors for 4 different vehicle types (two-wheelers, passenger cars, LDV, HDV). There is no distinction in type of tyre, brake or vehicle used. TIER 2 adds a speed correction based on Luhana (2004), reflecting higher emissions for both tyre and brake wear at lower speeds. Though this may seem counterintuitive for braking, the lower speeds in fact reflect driving conditions that require more and stronger brake inputs (e.g. urban stop-go) while on average having low speeds. Table 2 summarises the emission factors in the EMEP guidelines.

Table 2: EMEP TIER2 base emission factors for the different types of non-exhaust PM

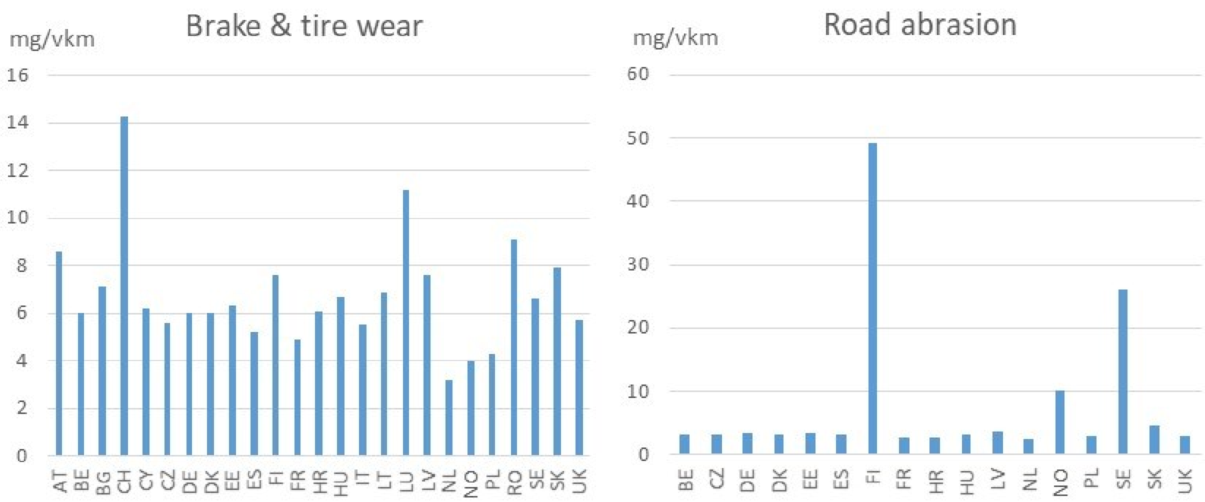
	Two-wheel vehicles TSP emission factor (mg/km)	Passenger car TSP emission factor (mg/km)	LDV TSP EF (mg/km)	%PM_{10}	%$PM_{2.5}$	%PM_{1}	%$PM_{0.1}$
Tyre wear	4.6	10.7	16.9	60%	42%	6%	4.8%
Brake wear	3.7	7.5	11.7	98%	39%	10%	8%
Road abrasion	6.0	15.0	15.0	50%	27%	unspecified	unspecified

Source: Ntziachristos and Boulter (2019).

The speed dependency is considered to be a linear one between 40-90 km/h, ranging from 1.4 at 40 km/h to 0.9 at 90 km/h with the base value at 80 km/h for tyre wear and ranging from 1.67 at 40 km/h to 0.185 at 95 km/h with base value at 65 km/h for brake wear.

Most EU countries use the EMEP/EEA Guidebook to develop national emission inventories. Indeed from the implied emission factors based on the available EMEP/EEA emission inventory data, we find most countries rely on this data for compiling their emission inventories (Van Der Gon 2018). **Error! Reference source not found.** summarises these findings, revealing few countries with significant deviations compared to the emission factors from the EMEP/EEA Guidebook. Switzerland (CH), Luxemburg (LU), and Romania (RO) seem to combine brake and tyre wear and road abrasion in a single value. Nordic countries (here Denmark, Norway, Sweden, and Finland) have a higher implied emission factor for road abrasion associated with the use of studded tyres. Finally, the Netherlands (NL), Poland (PL), and Norway (NO) use a different emission estimation methodology. The large differences in emission factors of road abrasion among the Nordic countries is somewhat puzzling, suggesting high uncertainty in the emission estimates.

Figure 2: Implied brake and tyre wear (left) and road abrasion (right) PM10 emission factors



Source: Van Der Gon (2018).

It is interesting to single out a few countries that deviate from the EMEP/EEA Guidebook methodology, such as the Netherlands. The Dutch emission inventory is described in Klein (2019) and relies on a literature review by Ten Broeke (2008). Emission factors are summarized in

Table 3.

Table 3: Emission factors for the different types of non-exhaust PM in mg per kilometre driven

	Two-wheel vehicles TSP EF (mg/km)	Passenger car TSP EF (mg/km)	LDV TSPEF (mg/km)	$\%PM_{10}$
Tyre wear	4.6	10.7	16.9	60%
Brake wear	3.7	7.5	11.7	98%
Road abrasion	6.0	15.0	15.0	50%

Source: Ten Broeke (2008)

Emission factors differ slightly from the EMEP/EEA Guidebook emission factors, yet the share of coarse particles is higher in the Netherlands compared to EMEP/EEA Guidebook

Another interesting group of countries are the Nordic countries. In Sweden, the methodology was updated from 2017 onwards, using a different methodology based on Mawdsley (2015). The Swedish inventory uses the SIMAIR model and includes resuspension. The EMEP/EEA Guidebook is somewhat ambiguous in terms of resuspension. It states resuspension should not be included in the reporting yet acknowledges it is hard to isolate and exclude the resuspension (Ntziachristos and Boulter 2019)([6]). The Swedish methodology, though technically not in line with the EMEP/EEA Guidebook, is explicit and transparent by incorporation resuspension. This will explain Sweden as an outlier, compared to other EU countries.

Finally, we review the emission factors for non-combustion emissions in The Handbook of Emission Factors for Road Transport (HBEFA - https://www.hbefa.net/e/index.html). HBEFA is an alternative to the COPERT methodology embedded in the EMEP/EEA Guidebook. Though the majority of the EU Member States use COPERT, some use HBEFA for formal emission reporting (e.g., Germany, Switzerland, Austria). The most recent methodological report (Notter 2019) relies on a review by During and Schmidt (2016) for the estimation of non-exhaust PM emissions.

HBEFA does not distinguish between BWP and TRWP and reports a single non-exhaust PM emission factor. HBEFA distinguishes between road types, reflecting variation in speeds and general driving behaviour which influences the emission factor. The values are summarised in Table 4 below.

Table 4: HBEFA PM_{10} emission factors for different vehicle types (mg/vkm)([7])

	Urban road	Rural road	Motorway
Passenger car / LCV	54	22	47
Bus	540	144	74
Motorcycle	13.5	5.5	11.75
HDV	540	144	74

Source: Notter (2019).

Insofar comparable, HBEFA emission factors are in the same order of magnitude, yet are higher compared to the EMEP/EEA Guidebook emission factors, likely because resuspension is implicitly

[6] In official emission inventories, re-suspension has explicitly not to be included in the national primary PM totals. However, e.g. the re-suspension of road-side PM can be reported separately as so-called memo item.
[7] vkm is a measure of traffic flow, determined by multiplying the number of vehicles on a given road or traffic network by the average length of their trips measured in kilometres.

included in the HBEFA emission factors as well. The HBEFA methodological report suggests resuspension is included, however a clear statement is lacking as all non-exhaust emissions are grouped in a single value.

The current typology and large variation both in terms of emission estimation approaches as well as the apparent variation in the emission factors, suggest there is large uncertainty on road transport non-exhaust particle emission estimates. The most important methodology for emission inventory estimation (EMEP/EEA Guidebook) is outdated. Most authors dealing with the topic, agree on the importance of non-exhaust PM emissions as a specific potential health risk and acknowledge the uncertainty of the current emission estimates.

2.2 Rail transport

Though the focus of this paper is on road transport, also other modes of transport generate non-exhaust emissions. Rail transport, be it trains, trams or metro, generate PM from abrasion of power lines, tyres on rails and brakes.

The EMEP/EEA Guidebook does not consider non-exhaust emissions of rail transport a separate category. Non-exhaust PM emissions are thus to be included in the 1.A.3.c category (railways)([8]). The EMEP/EEA Guidebook for the 1.A.3.c category of railways (Norris 2019) doesn't mention non-exhaust PM of railways, so reporting with respect to the emission inventory are not clear.

The German emission inventory includes an estimation of non-exhaust PM of rail transport (UBA, 2020). Relying on further unspecified input from the German railroad company Deutsche Bahn AG, the German emission inventory distinguishes between PM from the overhead power line, tyres on rails and the braking system. In case of railways, it's not only PM per se but the composition of the particles, i.e. heavy metals, which is of particular concern. Wear from overhead power lines are assumed to be 100% Cu. Particulates generated from friction between tyres and rails is assumed to be 100% iron (Fe) and braking wear is assumed to be a steel alloy containing chromium (Cr) and nickel (Ni). Table 5 summarises the emission factors.

Table 5: Emission factors for different types of rail non-exhaust PM (mg/vkm)

	$PM_{2.5}$	PM_{10}	TSP
Power line	0.16	0.32	0.32
Tyres on rails	9	1.8	1.8
Brakes	4	8	8

Source: UBA (2020)

Non-exhaust PM emissions were included from 2018 onwards in the German emission inventory and have led to a large increase of reported emissions under the 1.A.3.c category. The rail non-exhaust emissions now account for about 95 % of total PM and the rail emission category now accounts for 4% of the national total of Cu emissions (Hausmann 2020).

The Dutch emission inventory (Klein 2019) also includes wear of overhead powerlines and developed an approach based on Coenen and Hulskotte (1998) and CTO (1993), using emission factors of

[8] see NFR 14 (Nomenclature for reporting/UNECE nomenclature for reporting of air pollutants), Conversion chart for aggregated sector groups, in EEA, 2020 (https://www.eea.europa.eu/publications/european-union-emission-inventory-report-1990-2018).

15mg/kWh for overhead lines and 10 mg/kWh for carbon brushes. Due to the composition of 100% Cu in the overhead lines and 10% Pb in the carbon brushes, the emissions of power line abrasion alone account for 12.5% and 2.8% of the national total emissions of Cu and Pb respectively. It is assumed that 20% is released as PM of which most of it is deposited in the vicinity of the railway (Coenen and Hulskotte 1998).

There is particular concern on non-exhaust PM emissions in subway systems, given the closed environment, allowing particles to accumulate in the air on station platforms as well as inside carriages. Yingying (2016) concludes from measurements that particulate mass in subway tunnels increased by a factor 12 during braking conditions. The generated PM enter the carriages as train doors open and lead to increase PM concentrations of 5-25%. Martins (2014) similarly finds elevated PM levels at platforms, 1.3-6.7 times higher compared to an outdoor environment in the Barcelona subway system. Ventilation and use of air-conditioning can improve air quality inside the carriages.

Abassi (2013) concludes the generation of non-exhaust PM emissions in subways depends on the axle load, brake material and system as well as infrastructure and operational parameters such as rail gauge and vehicle speed (determining the intensity of braking). Given the health concern of elevated particle levels, limited cohort studies investigated adverse health effects on subway drivers, a cohort in principle most exposed to this specific type of non-exhaust emissions. Bigert (2007) and Gustavsson (2008) did not find any elevated frequencies of heart attacks and lung cancer respectively.

2.3 Air transport

As with rail, non-exhaust emission are not considered as a separate category in the EMEP/EEA Guidebook and are assumed to be included in the aircraft category 1a3.a([9]) category. There are 2 main sources of non-exhaust PM during landing and take-off (LTO): tyre wear on touchdown and braking action during taxiing (Walker 2018).

Only a limited amount of research into the non-exhaust PM emissions of aircraft was found. The Dutch emission inventory (Dellaert and Hulskotte 2017) takes into account both brake and tyre wear and employs emission factors of 0.223 g/ton Maximum Take-Off Weight (MTOW) and 0.253 g/ton MTOW respectively, citing Morris (2007) as source for the emission factors. Combined, tyre and brake wear account for 40% of total LTO PM emissions, i.e. also considering PM exhaust emissions.

Heathrow airport, one of the largest airports in Europe, compiled its own emission inventory and finds similar values with a share of 42% of the total ground-level PM emissions of aircraft (Walker 2018). Emission factors in the Heathrow emission inventory are exactly the same as the values used in the Dutch emission inventory, citing the project Icarus as a source (Peters 2009). It is assumed that 10% of the eroded mass of aircraft tyres is suspended persistently as PM_{10}. Though aircraft tyres differ from tyres from road vehicles, the same $PM_{2.5}/PM_{10}$ ratio is applied.

Symptomatic for the degree of uncertainty on tyre wear emission on touchdown, are findings of the National Academies of Sciences (2013), finding a large variation of tyre emission, even for similar aircraft, for example values of 2 to 71 mg of tyre material loss per landing for aircraft type B757. The authors stress 'the importance of the precise touchdown dynamics (which will depend on aircraft loading, weather conditions, and the pilot's response to the entire situation) can strongly influence the resulting emissions' (National Academies of Sciences 2013). Interestingly, the conclusions of the report in terms of the importance of tyre and brake wear in the total LTO PM emissions differ significantly

[9] 1A3ai(i) International aviation landing and take-off (LTO) (civil) and 1A3aii(i) Domestic aviation LTO (civil).

from the Dutch and Heathrow emission inventories. While the latter agree on a share of about 40%, the former estimated the share of tyre wear less than 1%.

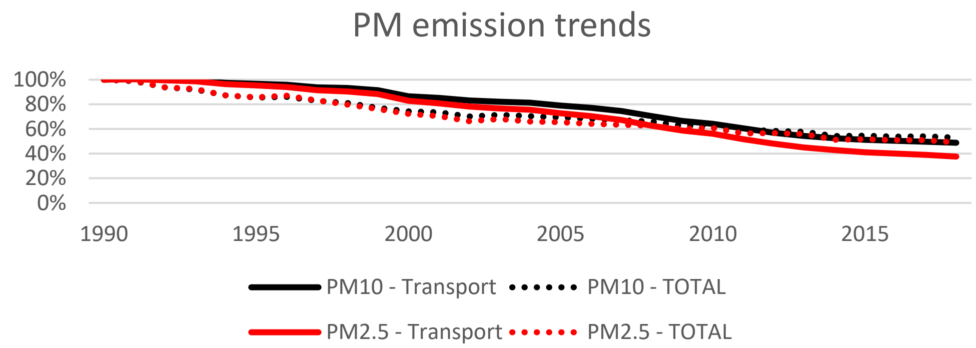
From the limited literature available, it is clear that large uncertainties remain with respect to non-exhaust emissions from aircraft in LTO.

3 Relevance and trends

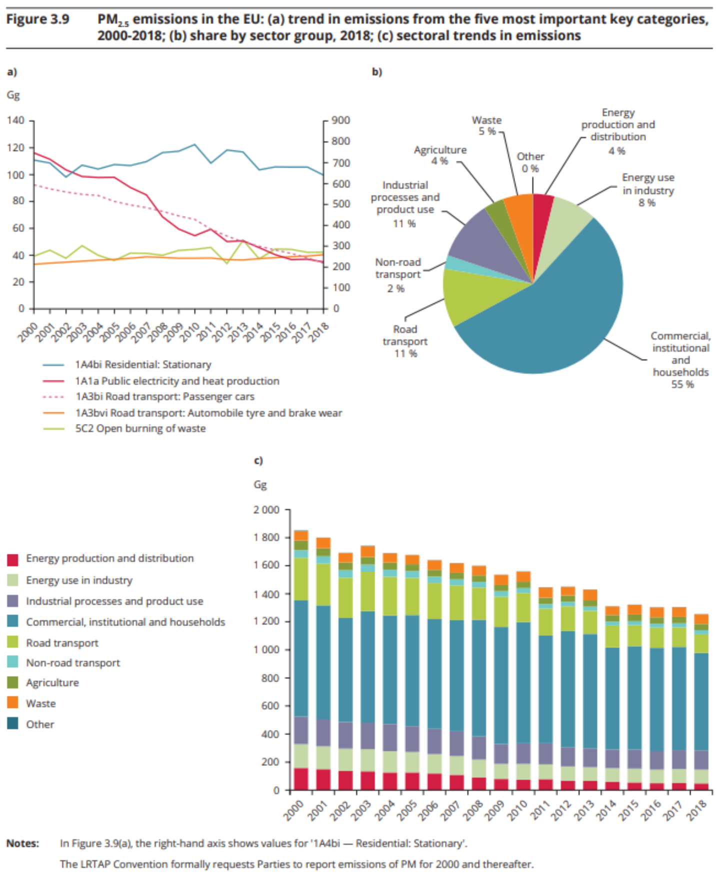
3.1 Historical trends

Total primary PM emissions to the air have been declining in the EU as mitigation policies have taken effect. Both for $PM_{2.5}$ and PM_{10}, the emissions of transport have gradually declined to about half the emission in 2018 compared to 1990. The transport sector has consistently taken a share of 12-20% of the total PM emissions over the past decades and now accounts for 12% and 13% of PM_{10} and $PM_{2.5}$ emissions respectively in the latest emission inventory (EEA 2020). Road transport accounts for about 11% of total primary PM emissions; other modes of transport account for about 2% of total primary PM emissions.

Figure 3: Relative primary PM emission trends for all sectors and transport in EU28



Source: EEA 2020 – data download September 2020.

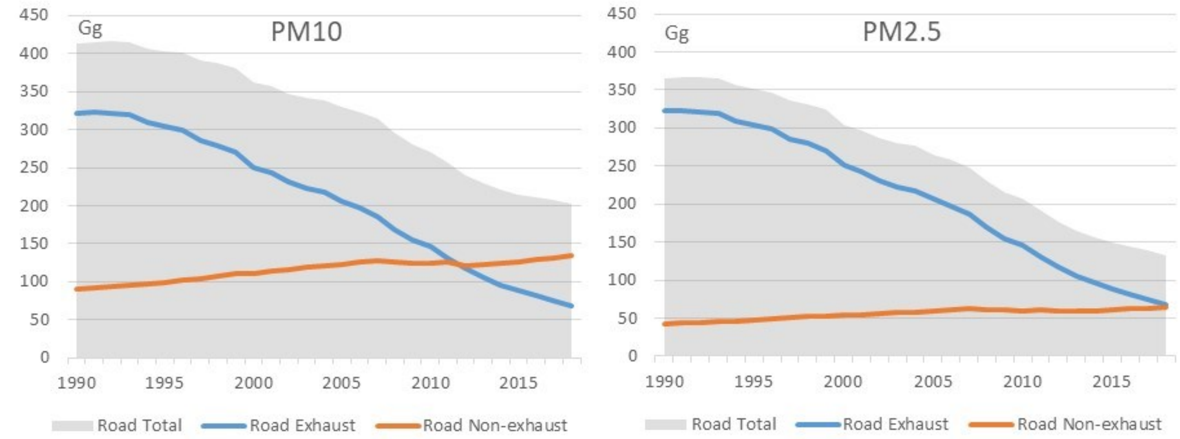


Source: 2020 LRTAP report.

Road transport is the dominant sector, typically accounting for about 85% of total transport PM emissions. The remainder of this chapter thus primarily focuses on the road transport sector.

While transport emissions are steadily declining, there is a major shift in the underlying sources. The reduction of PM emissions is driven by the reduction of exhaust emissions, while non-exhaust emissions have increased over time as transport activity has continued to increase without any mitigation measure in place specifically targeting non-exhaust PM emissions. This observation primarily holds for PM_{10}, as non-exhaust particulate emission are more coarse, overtaking exhaust emissions as the dominant source in 2012 for the PM_{10} fraction. As of 2018, also $PM_{2.5}$ non-exhaust emissions are on par with exhaust emissions. With the total PM emissions decreasing and non-exhaust PM emission from transport increasing, the share of non-exhaust PM in the total PM emissions has increased from 2% in 1990 to 7% in 2018 (EEA 2020).

Figure 4: PM10 (left) and PM2.5 (right) emissions from road transport in EU28 (1990 to 2018)

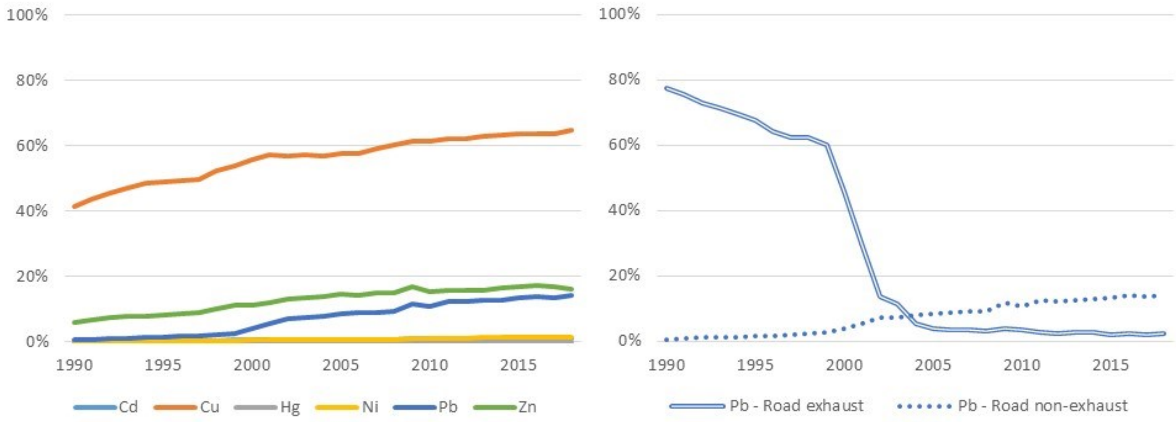


Source: EEA2020 – data download September 2020.

Zooming in on metals, the relative contribution of non-exhaust emissions from transport is found to be an important sub-sector for some pollutants and steadily increasing over time. In particular for Cu, the non-exhaust emissions from road transport alone are the majority of national total emissions at 65%. Also for Lead (Pb) and Zn, the importance of non-exhaust emissions have increase with relative shares to the national totals increasing from 1% in 1990 to 14% in 2018 and 6% in 1990 to 16% in 2018 respectively.

The case of Pb is symptomatic for the exhaust and non-exhaust emission trends in transport. At the end of the 20th century, Pb emissions from transport were well over half the total national emissions. The challenge of Pb emissions from transport was met by banning Pb as a fuel additive, drastically reducing the total Pb emissions. With exhaust emissions now under control by fuel standards, Pb is re-emerging as a concern from the non-exhaust emissions. While indeed slowly increasing again, Pb emissions are still well below the levels of 1990 at about 1.5% in 2018 compared to 1990.

Figure 5: Emission trends of selected metals emissions from transport, share of total emissions in EU28 (left) and in detail by sub-sector for Lead (right)



Source: EEA 2020 – data download September 2020.

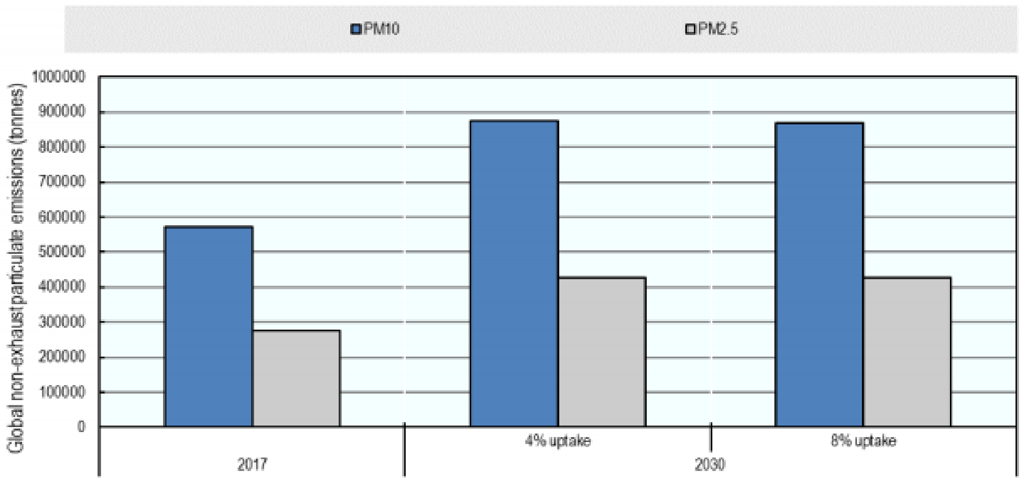
Bearing in mind that most countries are assumed not to report metal emissions from rail tyre and brake wear, in the way the Netherlands and Germany are including it in their emission inventory, the share of non-exhaust PM-emissions as a source for metal emissions might be higher still compared to what EMEP data reveals.

3.2 Projections

The historical emission trends diverge for exhaust emissions (declining) and non-exhaust emissions (increasing). In the absence of effectful mitigating measures, this trend is expected to persist in the future. Two trends are relevant: [1] the constant, gradual fleet renewal, replacing old and polluting road vehicles by new vehicles that comply to more stringent emission standards and [2] the projected steady increase of transport demand, estimated at +/- 1% per annum for passengers and 1.5% per annum for freight between 2020 and 2030 according to the EU reference scenario of 2016 (Zampara 2016). The former reduces the average emission factor of exhaust PM while the latter leads to increasing non-exhaust PM emissions, as without any measures in place, emissions increase with traffic demand.

In a recent report, the OECD (2020) projects future global non-exhaust $PM_{2.5}$ emissions from road transport to increase by a factor of about 1.5 in 2030 compared to 2017, regardless of the uptake of electric vehicles in the fleet. This simulation demonstrates that sources of PM-emissions are expected to further shift towards non-exhaust sources and the gradual uptake of EV's, further reducing exhaust emissions only, will not alleviate the problem.

Figure 6: Emission rate estimates for the year 2017 and projected 2030 under alternative



Source: OECD, 2020.

Van der Gon (2018) reports on findings from Hulskotte and Jonkers (2012), revealing the growing importance of non-exhaust PM emissions from passenger cars and their contribution to PM levels in ambient air. Hulskotte and Jonkers (2012) used a simple air quality modelling approach applying the CAR-model (Jonkers 2007) and compared the share of non-exhaust PM in 2010 vs. 2020. Given the uncertainty on the non-exhaust emissions, the authors opt to estimate for 2 wear scenario's and for 3 different traffic conditions.

Table 6: Modelled PM_{10} levels in ambient air under alternative wear scenarios (mg/m^3 and between brackets break and tyre wear share in %)

	Standard wear scenario		**High PM_{10} fraction wear scenario**	
	2010	**2020**	**2010**	**2020**
Urban free flow	3.5 (49%)	2.8 (76%)	5.4 (66%)	4.9 (86%)
Urban intermediate flow	3.0 (48%)	2.2 (74%)	4.5 (65%)	4.0 (85%)
Congested traffic	4.2 (38%)	2.7 (66%)	5.7 (54%)	4.6 (80%)

Source: Hulskotte (2012).

Already in 2010, non-exhaust PM emissions are an important contributor to poor local air quality; in 2020 the contribution is increasing, taking the majority in all scenario's.

It is evident that as cleaner vehicles continue to replace old polluting vehicles, the importance of non-exhaust PM emissions will continue to increase with no effective mitigation measures in place.

3.3 Relevance of electric vehicles

Of particular interest for the future of non-exhaust emissions, are electric vehicles (EVs). EVs can be further broken down in Plug-in Hybrid Electric Vehicles (PHEV), full Battery Electric vehicles (BEV) and Fuel Cell Electric Vehicles (FCEV). PHEV's combine a conventional drivetrain with an (externally charged) battery powered electric motor. BEV and FCEV fully rely on external electricity (BEV) or hydrogen (FCEV) supply. BEV's and FCEV's eliminate all exhaust emissions and as such only generate non-exhaust emissions. For PHEV's, this only holds for when driving electric.

EV sales are expected to accelerate in the next decade, reaching a (sales) market share in excess of 30% in 2030 (IEA 2020).

It is important to recognise the impact of some specific characteristics of EVs on their expected non-exhaust PM emissions. There are 2 effects at play: [1] EVs combine regenerative braking[10] and friction braking versus internal combustion engine vehicles (ICEVs) relying solely on friction braking (Luin 2019), EVs thus avoid brake wear emissions and [2] EVs are heavier compared to ICEVs due to the need for battery's (Berjoza and Jurgena 2017), thus implying stronger brake wear (when friction braking is applied) and more tyre wear, as tyre wear correlates with vehicle weight (Pohrt 2019; Timmers and Achten 2016).

There is no scientific consensus as to what the net effect is. 'The net balance between reductions in brake wear emissions and potential increases in tyre and road wear emissions and resuspension for vehicles with regenerative braking remains unquantified' (Air Quality Expert Group 2019).

Timmers and Achten (2016) are a frequently cited source, and conclude that the combined exhaust and non-exhaust PM emissions of EVs are similar to the ones of ICEVs. The limited difference in non-exhaust PM from EVs versus ICEVs is mostly attributed to resuspension, which holds a high share in this study (50-75%). Resuspension is assumed to be higher for EVs due to higher weight. Under the given assumptions, when excluding resuspension, EVs would have 30-40% lower non-exhaust PM from tyre & break wear. The analysis seems limited as it assumes no brake wear emissions for EVs, which would imply EVs do not use friction braking, contradicting for example Luin (2019).

Hooftman (2016) considers non-exhaust PM emissions in an overall comparison of environmental performance of EVs versus ICEVs. EV tyre wear is assumed to be 10% higher while brake wear is assumed to be 33% lower, citing data revealing longer intervals between brake pad replacement for EVs. The author further assume no difference in road abrasion between EVs and ICEVs and conclude EVs have less non-exhaust PM emissions compared to ICEVs. As with (Timmers and Achten 2016), the analysis is limited and the conclusions are subject to strong assumptions.

Specific research into the topic is limited. Kendrick and Kulkarni (2019) estimate material loss of brake pad and disc in a lab environment in the common world harmonized light-duty vehicles test procedure (WLTP) drive cycle and modified WLTP cycle reflecting the use of regenerative braking and less friction braking for an EV. The authors found about 50% less material loss in volume (i.e. in mass concentration measured e.g. in $\mu g/m^3$) and 30% lower concentration of $PM_{2.5}$ number (i.e. measured in numbers per m^3), confirming assumptions in Hooftman (2016).

A promising analysis is done by Beddows and Harrison (2020), building on a model establishing a quantitative relationships between the PM emission factor and vehicle mass for tyre and brake wear and road abrasion individually. The authors find that EVs have a 258 kg to 318 kg higher vehicle mass

[10] During generative breaking, an electric motor is functioning as an electric generator. In battery electric (and hybrid) electric vehicles, the resulting kinetic energy is (partly) used and stored chemically (i.e. chemical energy) in a battery.

compared to diesel and petrol vehicles, respectively. When assuming 100% friction braking, EVs generate 7-12% more PM emissions, both from increased tyre and brake wear as a result of the heavier mass of the EV. Regenerative braking can effectively reduce overall emissions from EVs below ICEV levels. By virtue of avoiding brake wear, especially in the urban environment, this can be up to a 12% reduction compared to ICEVs. However, regardless of the potential of regenerative breaking to reduce brake wear, on motorways, EVs have higher PM emissions compared to ICEVs due higher tyre wear associated with higher weight.

Reflecting the need for further research, Beddows and Harrison (2020) conclude: ‘There are still very high uncertainties which overshadow these findings’.

4 Relevance for micro-plastics

There is a growing awareness that emissions of non-exhaust particles are an important source of microplastic to the environment (Magnusson et al., 2016; Sommer et al., 2018; Evangeliou et al., 2020; Baensch-Baltruschat et al., 2020). However, there is no consensus on the extension of what constitutes microplastic, as non-exhaust particles have substantial variability in properties (Sommer et al 2018). An additional challenge is that measurements need highly specialised equipment and consequently there are very sparse measurements of microplastic particles (MP) in ambient air pollution. In one of the few available measurement campaigns, Sommer et al. (2018) found, based on chemical and mineralogical analysis of traffic related abrasion particles, that particles are generally not pure tyre, brake or road particles but a composite of the various types of materials. As the composition of non-exhaust particles consists of a diversity of chemical compounds that, to a large extent, reflect the materials involved in the frictional process responsible for the wearing off, they seldom exist as pure particles. These features pose a challenge on the specification of microplastic from non-exhaust traffic emissions. Therefore, in order to understand the source of microplastics and establish the relevance of non-exhaust traffic emissions on their production, the materials subjected to wearing processes need to be well understood.

Hartmann et al. (2019) showed how ambiguous the term “microplastic” can be and, depending on field of science, the definitions vary in literature. The term is commonly applied to variable sizes, and this may be a source of uncertainty when comparing different studies. In their proposed framework, Hartmann et al. (2019) suggest that the size range of particles <1 µm is “nano” and <1 mm “micro”. As plastic, Hartmann et al. (2019) suggest an International Union of Pure and Applied Chemistry (IUPAC) definition, somewhat broader than the International Organization for Standardization (ISO) standard (ISO 472:2013): “Molecule of high relative molecular weight, the structure of which essentially comprises the multiple repetition of units derived, actually or conceptually from molecules of low relative molecular mass”. By this definition, most of non-mineral, non-metallic, non-exhaust emission particles would have a MP component, however, only particles with an aerodynamic diameter smaller than about 10 µm would be relevant as air pollutants. Microplastic air pollution is, therefore, a somewhat ambiguous term, which is poorly constrained by both size and chemical composition in literature.

The wearing rate of each of the involved constituents, i.e., road, tyre or brakes, varies with local road properties and conditions and different driving cycles. Thus, road surface properties, vehicle speed and weight, frequency and intensity of braking all play into which material is worn, and the amount, size and composition of worn particles that ultimately is suspended. In the following sections we go into the details of each of the materials and processes, assessing the microplastic component and its relevance for the proliferation of microplastic to the environment through atmospheric dispersion.

For suspendable microplastic production, tyre wear particles (TWP) are the most important, as a large part of the worn material is rubber tread and can therefore potentially produce the most microplastic. Bitumen is generally not considered a microplastic, but road wear particles (RWP) can also be a source of MP due to the presence and wear of thermoplastic road paint. Therefore, the characterization, both morphological and chemical, of the constituents of all WP is essential to determine the contribution to emissions of airborne microplastic, and ultimately to the microplastic share in PM_{10} or $PM_{2.5}$ air concentration levels. These aspects are summarised in the following subsections.

4.1 Microplastic air emissions

Global MP emissions have recently been estimated independently both by NILU and IIASA and published jointly in Evangeliou et al., (2020), with similar total results. Table 7 shows estimates from NILU's emission inventory for EU-28 countries plus Norway, along with a compilation of MP emission estimates done by Baensch-Baltruschat et al. (2020) for some of the countries. We have compared MP emissions with total particle emissions officially reported by each of the parties to the Convention on Long-range Transboundary Air Pollution (CLRTAP) for the road transport sector also visualised in Figure 7 . The derived share of MP of total road transport and non-exhaust emissions are shown in Table 6, and are estimated based on microplastic emissions (Evangeliou et al., 2020) and the official emissions reported to CLRTAP by the countries for 2015.

Based on the comparison between modelled MP emissions (Table 7) and reported traffic emissions, we have estimated that the share of microplastic of total emissions vary between 6 - 56% of total traffic emissions, and constitute 25% of the total traffic emissions in EU-28 plus Norway (Sum in Table 7). The comparison with non-exhaust emissions shows that microplastic emissions vary between 6 and 90% of the non-exhaust emissions, and constitute a total of 38% of the total non-exhaust emissions in EU-28 plus Norway (Sum in Table 7).

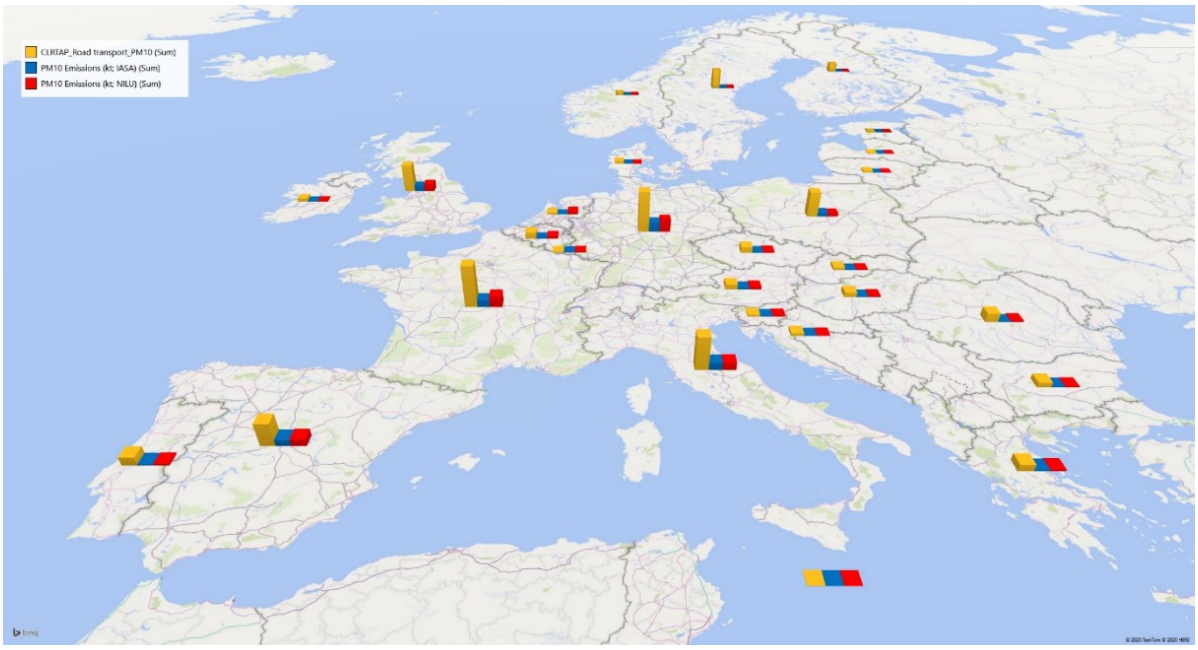
A factor that may affect these values is the relative contribution of the different emission sources. For instance, in Sweden and Finland, where studded tyres are widely used in winter, the contribution of road abrasion to emissions is higher than in other countries without widespread use of studded tyres. This results in a lower relative contribution of microplastic to non-exhaust emissions, i.e., 6 and 9%, respectively. The wide range of values may also in part be associated to differences in methodologies behind the estimate of non-exhaust emissions, but also probably reflects large uncertainties in both estimates. NILU emissions inventories for MP are based on a uniform approach for all countries, whereas the emissions reported by the countries are estimated by using their national methods.

Microplastic emission in the $PM_{2.5}$ fraction are in the emission inventories for MP estimated to be 10% of the tyre and brake PM_{10} emissions (Evangeliou et al. 2020). Taking this into account, the contribution of microplastic to total $PM_{2.5}$ emissions from the road transport is much lower than for PM_{10}, between 1 and 8% (4% of the total traffic $PM_{2.5}$ emissions in EU-28, Norway and United Kingdom), and between 3 and 19% of the non-exhaust $PM_{2.5}$ emissions (8% of the total non-exhaust traffic $PM_{2.5}$ emissions). In part, the lower share of total particle mass MP has for $PM_{2.5}$ than for PM_{10} is associated with the size range of exhaust particles, which generally has almost all its mass within the $PM_{2.5}$ range, and that tyre particles have a weight in a coarser fraction of particles. It may also be affected by the coarse approximation that 10% of the tyre and brake PM_{10} emissions.

Table 7: PM_{10} microplastic particle emissions estimated by NILU (Evangeliou et al., 2020), and emissions compiled by Baensch-Baltruschat et al., (2020). Share (%) of MP in total PM_{10} emissions from traffic and in non-exhaust emissions.

Country	MP Emissions NILU	MP Emissions (Baensch-Baltruschat et al., 2020)	MP in PM_{10} total Traffic Emissions (%)	MP in PM_{10} non-exhaust emissions (%)
AT	1.75		41.77	85.02
BE	1.93		35.06	61.48
BG	0.61		18.10	50.04
CY	0.09		19.45	40.79
CZ	1.35		30.59	54.06
DE	9.03	13.3	28.51	42.64
DK	0.63	0.7	25.20	38.41
EE	0.14		25.45	50.16
EL	0.90		23.45	48.01
ES	4.11		27.08	48.01
FI	0.64		8.16	9.53
FR	7.43	3.76	22.89	49.02
HR	0.57		30.19	68.96
HU	0.83		24.35	50.12
IE	0.62		22.34	38.32
IT	4.82	5	19.62	34.85
LT	0.29			
LU	0.71			
LV	0.21		26.72	57.05
MT	0.02		11.03	25.02
NL	2.34	1.73		
NO	0.65	0.85	24.64	35.04
PL	2.57		18.56	37.37
PT	0.91		17.41	47.35
RO	0.98		18.86	44.55
SE	0.99	1.3	6.07	6.36
SI	0.65		54.00	
SK	0.55		29.55	58.19
UK	5.69	6.3	23.88	37.66
Sum	51.98		23.88	40.39

Figure 7: Total PM10 emissions from road transport submitted to the CLRTAP



Source: Author's compilation based on data from Centre on Emission Inventories and Projections, (https://www.ceip.at/), and microplastic particle emissions produced by IIASA and NILU (Source: Evangeliou et al., 2020).

4.2 Microplastic fraction in PM10 levels

Assuming that microplastic particles mainly come from tyre use and wear processes, knowledge on the contribution from TRWP and TP (see Box 1) to air pollution is essential. Few studies report measured MP from roads in an urban environment, and they generally conclude that the share of TWP in PM_{10} levels is relatively low. Panko et al. (2018) quantified, based on the pyrolysis marker, that TWP concentration in the PM_{10} fraction is relatively low, representing 0.84% on average contribution to total PM_{10} levels. According to the same authors in an urban study, the TP share of PM_{10} air concentration was found to be around 50% of the TRWP mass levels, and the average contribution to PM_{10} at urban locations was found to be 0.45-2.48% of TRWP and 0.23-1.24% of tread particles (Panko et al., 2019).

As the contribution of the different traffic emissions sources to PM_{10} concentration levels in ambient air depend on several factors, such as location and the distance to the different emission sources, seasonality and local and regional meteorological conditions, among others, it is not clear that MP from vehicles poses a significant challenge to air pollution. Whereas several studies confirm that part of the non-exhaust pollution is microplastic, the studies generally point to a minor contribution (Panko et al., 2018; 2019). It is possible to assume that microplastic particles generally constitute a lower share of total PM_{10} concentration levels than the share indicated by modelled emission (e.g., Table 7).

The relative importance of microplastic production from tyre wear process to air pollution will be to a large extent determined by their impact on human health and the environment. MP particles have been shown to be relatively inert when digested (Sommer et al. 2018), but inhalation poses different questions that need to be answered. These questions have only recently been investigated in depth and there is very thin evidence in the literature and the available results show contradictory results (see review Grikorator and Martini 2014).

With exhaust emissions declining, mainly due to more demanding regulatory requirements and technological advances, and non-exhaust emissions remaining at relatively steady level, road traffic is still one of the largest local sources of (primary) PM in most European cities. The amount of tyre particles produced per year ranges from 0.2 and 5.5 kg per capita in Europe (Baensch-Baltruschat et al. 2020). The available data make this a robust estimate on the order of magnitude of emissions. However, there are still large uncertainties such as determining the fraction of the MP mass, especially from tyres, that is available for resuspension (Evangeliou et al., 2020).

5 Technical mitigation options

Technical options are available to reduce the impact of non-exhaust emissions (NEE) as a source for poor air quality or microplastics in the environment. The available options can broadly be categorized as follows:

- Reducing formation of particles;
- Trapping particles at the source after formation;
- Removing particles from the environment.

5.1 Reducing formation

With respect to the generation of BWP, brake system design and material use are highly influential for the generation of particles. Gramstat (2018) and underlying sources provide a comprehensive review of brake materials used and relevance for the generation of PM. Common approaches are the use of coating on the brake disc to improve thermal conductivity. Increasing thermal conductivity is aimed to avoid high disk temperature (as a result of strong braking input) as particle generation strongly increases over the 'transition temperature' of 170°C (Perricone 2018). Coating technology is also used to harden the surface of the brake disc and, thereby, reduce wear and particle formation. High-Velocity-Oxy-Fuel (HVOF) coating is one such technology. Apart from coating, thermochemical treatment of the brake disc such as ferritic nitrocarburizing (FNC) is an alternative to improve durability and avoid wear. Avoiding cast iron or steel discs and use carbon-ceramics instead, can reduce disc wear but comes at a much higher cost. Gramstat (2018) further stresses the potential to avoid brake wear in particular on the side of the brake linings, suggesting possible copper-free alternatives, yet warns for adverse impact on friction performance.

Overall brake performance, increasing durability of the brake system and avoiding brake wear are aligning objectives. Any improvement on thermal conductivity to improve brake performance and durability will create a positive spill-over on the generation of brake wear particles. Gramstat (2018) concludes several technical solutions exist to further improve beyond current performance but cites cost and trade-offs with other characteristics (e.g. noise) as a barrier.

Another technological option to reduce BWP generation, is the use of regenerative braking. Regenerative braking recovers the kinetic energy of the moving vehicle into an energy carrier for reuse when accelerating. The use of regenerative brakes is more common in EV's as the electric motor can operate as the electric generator and electricity storage is available through batteries. Regenerative braking will always need to be supplemented with friction brakes as the braking effect reduces at low speeds and is insufficient to bring a vehicle to a full stop. Beddows and Harrison (2020) assume regenerative braking will lead to a reduction of 90% of use of friction brakes on EV's, citing real world observations in Los Angeles (Hall 2017). Hooftman (2016) is more cautious in the impact of regenerative braking, yet still assumes a 66% reduction for EV's compared to ICEV's, solely attributed to the use of regenerative braking. Gramstat (2018) concludes use of regenerative braking reduces disc temperature by 48% implying large potential to avoid brake wear.

Similarly, to avoid the generation of TWP, the material composition of the tyres is the primary focus.
Harper (2017) explores use of different polymeric materials while Grigoratos (2018) explored characteristics aiming to find a correlation between the treadwear rating of the tyre and particle generation. While no correlation could be established, authors found large variance between 55 and 212 mg/km for tyres of different brands, suggesting there is potential in tyre design and material use, aiming for low wear. Specifically for Nordic countries, specification of studded tyres can ensure low TRWP generation. Gustafsson (2017) shows that tyres with more studs generates higher PM_{10} levels.

An obvious measure to avoid RWP (and indirectly TRWP) is to have a road network with a rigid surface., avoiding susceptible surfaces like gravel (Penkala 2018). However, rigid road surfaces contribute to soil sealing which puts biodiversity at risk and increases the risk of flooding and water scarcity.

As vehicle weight influences both brake and tyre wear, reducing vehicle weights will impact the generation of both BWP and TRWP. Beddows and Harrison (2020) finds that both the higher BWP and TRWP of EV's, 6.8 – 11% compared to ICEV's, increase proportional to the higher EV weight increase of about 24% compared to ICEV's. This relation holds true for all vehicle types, regardless of the propulsion technology.

Driver assistance systems, aiming to reduce strong brake inputs by the drive and facilitate smooth driving to reduce tyre wear are an indirect way to reduce particle formation. OECD (2020) provides a summary of potential applications without estimating the emission reduction potential.

Finally, the air quality expert group (2019) in the UK and EEA (2018) also formulate non-technical measures such as traffic management, aiming for smooth traffic flows that lead to driving behaviour requiring few brake inputs, to avoid generation of BWP and TRWP. In particular in urban conditions, frequent stop and go traffic give rise to elevated BWP.
Interestingly, the authors also advise on speeds reduction on where traffic is free-flowing, for example on motorways, while Ntziachristos and Boulter 2019 citing Luhana (2004) expect TWP emissions to increase on a per km basis. Lower speeds in free flow conditions, will likely not per se lead to lower TWP generation, but will indeed lead to improved ambient PM concentrations due to lower levels of resuspension (Querol 2018).

5.2 Trapping at the source after formation

Apart from reducing formation, technological options exist to trap and collect generated particles at the source immediately after generation. Research into trapping BWP seems to be more evolved with some technologies tested in lab and real world environments. Hascoët and Adamczak (2020) present results from the TAMIC system by Tallano, an automotive supplier. The Tamic system core is an aspiration system to collect brake particles and consequently collect particles with a high efficiency filter. The impact on BWP emission reduction both in mass and number are tested in a lab environment, claiming 85% mass and 90% particles collected in the system. The system has no adverse effect on vehicle performance and braking capacity.

Mann-Hummel, another automotive supplier has developed a similar system, using a high efficiency filter. A passive system, the filter is installed near the brake pad and absorbs generated particles. Mann-Hummel claims 80% of the generated particles are captured in the system (Brake report 2018).

A similar wear particle collection device is under development for tyre dust by the Tyre Collective[11]. Imperial College London published a press release, including a 'claim that their prototype can collect 60% of all airborne particles from tyres, under a controlled environment on their test rig'[12], however we could not find any scientific documentation to substantiate these claims.

[11] https://www.thetyrecollective.com/.
[12] https://www.imperial.ac.uk/news/204514/tackling-harmful-tyre-emissions-student-inventors/.

5.3 Removal from the environment

Finally, it is also possible to remove particles from the environment to avoid resuspension. Available options are street sweeping, street washing with water or use of chemical dust suppressants.

The effectiveness of street sweeping without water is disputed due to the fact this technique is unable to catch particles under 10 µm. Querol (2018) provides a comprehensive summary of the benefits and drawback of street sweeping. Though likely ineffective to reduce Pm directly and in fact leading to short term elevated PM_{10} concentrations due to resuspension, the removal of large particles is considered beneficial, because it avoids generation of smaller particulate matter.

Water washing increases the effectiveness of classic (dry) street sweeping, but is equally unlikely to address PM_{10} directly (Querol 2018).

Querol (2018) summarizes 'The efficacy of road washing depends highly on several factors such as (1) climatic conditions, (2) road dust loadings, (3) frequency of washing, (4) road surface material, (5) portion and length of the road that is washed, and (6) relevance of other sources of PM.' And concludes the impact on curbside concentrations of PM_{10} attributed to street washing is 6-18.5%, citing a range of experiments. Effects are short-lived; in one example, the effect waned of a 18.5% reduction of PM_{10} concentrations immediately after sweeping, to a mere 2.2% reduction one day later (Amato 2016).

6 Policy options to mitigate non-combustion emissions

6.1 EU legislation

There are two main types of EU legislation. Directives set out the goal that all EU Member States must achieve but does not prescribe the concrete means of achieving it. Each Member State must then implement the Directive at national level. Regulations are passed at EU level, and they have direct effect in every EU Member State.

There is no EU legislation that directly addresses non-exhaust emissions (NEE) from vehicles. However, NEE are (indirectly) addressed e.g. in the EU's air pollution legislation:

The **National Emission reduction Commitments (NEC) Directive** (EU, 2016), covers primary $PM_{2.5}$ emissions and sets EU and Member State emission reduction commitments for 2020 and 2030. Reporting of primary PM_{10} as well as black carbon (BC) emissions are covered by the Directive, too, but the legislation does not set any reduction commitments (EEA, 2020b). The compilation and reporting of primary PM emissions must include NEE sources of PM from tyre and brake wear and road surface wear (road abrasion). In official emission inventories, countries do not have to include re-suspension in their national totals for primary PM. However, e.g. the re-suspension of road-side PM can be reported separately as so-called memo item.

The EU **Ambient Air Quality (AQ) Directive** of 2008 (EU, 2008) covers PM mass concentrations in ambient air (i.e. also released by non-combustion sources). The Directive sets air quality standards for the protection of human health (limit and target values) for PM_{10} and $PM_{2.5}$. The AQ Directive of 2004 (EU, 2004) is also relevant with respect to NEE since it covers heavy metal concentrations in the air (please see summary in EEA's annual AQ report ; EEA, 2020).

Other examples of EU Directives addressing also non-exhaust emissions by banning the use of certain toxic substances from motor vehicles and their components are:

- the **End of Life Vehicles Directive** (EU, 2000), which requires that "Member States shall ensure that materials and components of vehicles put on the market after 1 July 2003 do not contain Lead (Pb), Hexavalent chromium (Cr VI), Cadmium (Cd) and Mercury (Hg)", and
- The **Braking Devises Directive** (EU, 1998) enforced, which enforced from 1999 onwards asbestos-free brake pads for all road vehicles, which has an impact on the chemical composition of non-exhaust PM emitted to the air.

Chapter 5 summarises technical measures for mitigating NEE. Such measures can be reflected in EU Regulations. These can require specific approval processes. An example is the **Tyre Labelling Regulation** (EC, 2018). It includes a provision for future labelling as soon as a test standard is set: «... The mileage of tyres is related to their durability and life expectancy. Tyre abrasion is a major source of microplastics released into the environment. No test has yet been developed that would allow the mileage or abrasion rate of tyres to be measured reliably. It is therefore proposed to consider using delegated powers to include these parameters in the future, once an appropriate test standard is finalised....».

Worth mentioning are also **REACH**([13]) (EU, 2006) and REACH-like Regulations, the European Regulation on the classification, labelling and packaging of chemical substances and mixtures (**CLP Regulation**) (EU, 2008b) as well as Regulations related to restrictions on the use of particular trace elements and heavy metals (e.g., Grigoratos, 2018; OECD, 2020).

[13] REACH = Registration, Evaluation, Authorisation and Restriction of Chemicals.

Another example, indirectly related to non-exhaust emissions, is a regulation addressing **labelling of tyres with respect to fuel efficiency** (EU, 2020).

6.2 "Avoid, shift, improve" – sustainable mobility system

A broader approach to reduce (also) non-exhaust emissions from road traffic - not addressed in this report – is reducing vehicle-kilometers travelled. The authors of a study published by the Organisation for Economic Co-operation and Development (OECD, 2020) argue: "given that vehicle travel entails a range of other negative externalities (e.g. congestion and greenhouse gas emissions), reducing the number of vehicle-kilometers travelled, especially in urban areas, should be a key component of policy portfolios to mitigate non-exhaust emissions." The authors suggest economic instruments, i.e. policies that focus on disincentivising the use of private vehicles and incentivising the use of alternative modes such as public transport, cycling, and walking. They state that the "avoid, shift, improve" conceptual framework, which addresses the current mobility system in general, should also be applied to non-exhaust emissions (see e.g. also EEA, 2018a and EEA, 2019). Policies that incentivise higher occupancy levels of private cars can also be suitable tools for reducing NEE.

Box 3 - Disincentivises and incentivises

> "Disincentives for private vehicle ownership and use include pecuniary measures, e.g. registration fees, fuel taxes, distance-based charges, and parking pricing, as well as regulatory measures, e.g. urban vehicle access regulations and other types of vehicle bans. Incentives to increase the uptake of alternative modes include improving the coverage, frequency, comfort, information provision, and payment systems of public transit services and improving the quality and coverage of infrastructure for non-motorised modes, such as protected bike lanes, sidewalks, and priority pedestrian crosswalks. In the long term, developing compact urban areas can also contribute to reducing demand for private vehicle use by shortening the distances required to access amenities." (OECD, 2020)

Further, OECD (2020) refers to urban vehicle access regulations (UVARs) as a means for reducing (also) non-exhaust emissions. Examples are low-emission zones and congestion pricing schemes (see e.g. EEA, 2013; EEA, 2018b).

7 Conclusions

Exhaust PM emissions have been decreasing over the last 20-30 years, under impulse of targeted policy interventions. Meanwhile, non-exhaust PM emissions from transport are steadily increasing on par with increasing transport demand. In particular sources of non-exhaust PM emissions from road transport, be it from brake, tyre or road wear have all increased to a point non-exhaust PM emission have overtaken exhaust emissions as the dominant emission source in transport as of 2012 for PM_{10} and 2018 for $PM_{2.5}$. Also for rail and aviation, there is emerging evidence on the importance of non-exhaust emissions.

While exhaust emission inventories can rely on established guidelines, building on extensive scientific research, the estimation of non-exhaust emissions is subject to large uncertainty. There is a need to reduce uncertainty and improve the emission inventory guidelines, incorporating the latest scientific evidence in order to tailor an adequate policy response.

Apart from non-exhaust PM emissions' relevance for air quality, there is emerging evidence that non-exhaust PM from road transport are an important source of microplastics. Currently, there is limited evidence on the environmental and health impacts, signalling a need for further research and an additional argument to craft a coordinated policy response to reduce non-exhaust emissions.

At the policy level, historically the focus has almost exclusively been on reducing exhaust-emissions. Few policy interventions currently exist that affect non-exhaust emission directly. Technical solutions are available to substantially reduce non-exhaust PM, in particular for road brake and tyre wear. A variety of regulatory options are available such as emission limits, tyre and brake standards.

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9 Abbreviations

Al	Aluminium
BC	Black Carbon
BWP	Brake Wear Particles
Cu	Copper
EF	Emission Factor
EMEP	European Monitoring and Evaluation Programme
EU	European Union
EV	Electric Vehicle
Fe	Iron
FNC	Ferritic NitroCarburizing
HBEFA	Handbook Emission Factors for Road Transport
HDV	Heavy Duty Vehicle
ICEV	Internal Combustion Engine Vehicle
ISO	International Organization for Standardization
IUPAC	International Union of Pure and Applied Chemistry
K	Potassium
LACT	Los Angeles City Traffic
LCV	Light Commercial Vehicle
LDV	Light Duty Vehicle
LTO	Landing and Take-Off
Mg	Magnesium
MP	Microplastic Particles
MTOW	Maximum Take-Off Weight
NAO	Non-Asbestos Organic
NEE	Non Exhaust Emissions
NFR	Nomenclature For Reporting
PAH	Polycyclic Aromatic Hydrocarbon
Pb	Lead
$PM_{0.1}$	Particulate Matter with a diameter of 0.1 µm or less
PM_1	Particulate Matter with a diameter of 1 µm or less
$PM_{2.5}$	Particulate Matter with a diameter of 2.5 µm or less
PM_{10}	Particulate Matter with a diameter of 10 µm or less
RWP	Road Wear Particles
Sb	Antimony
Si	Silicon
Sn	Tin
TP	Tread Particles
TRWP	Tyre and Road Wear Particle
TSP	Total Suspended Particles
TWP	Tyre Wear Particles
vkm	Vehicle kilometres
WP	Wear particles
Zn	Zinc
Zr	Zirconium

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